Global Ecology and Conservation 4 (2015) 160–175

Contents lists available at ScienceDirect

Global Ecology and Conservation

journal homepage: www.elsevier.com/locate/gecco

Original research article

How will fisheries management measures contribute towards the attainment of Good Environmental Status for the North Sea ecosystem?

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HIGHLIGHTS

- We modify effort by fishing fleets to meet goals for ecosystem provisioning.
- We model wider effects of fishing on ecosystem structure and functioning.
- We show how indicators of biodiversity and food webs will respond to pressures.

ARTICLE INFO

Article history:
Received 24 April 2015
Received in revised form 26 June 2015
Accepted 26 June 2015
Available online 6 July 2015

Keywords:
Temperature
Ecopath with Ecosim
Ecosystem based management
Mixed fisheries

ABSTRACT

European fisheries management have adopted maximum sustainable yield (MSY) targets for fishing mortality on commercial species to maximise the provision of food. EU member states are also committed to reach Good Environmental Status by 2020 through the Marine Strategy Framework Directive which aims to protect all ecosystem services. So, how will fisheries management measures contribute to ecosystem functioning and the attainment of environmental objectives as measured by improvements in indicators of biodiversity and food webs? We model ecosystem effects of fishing in the North Sea using food web model projections incorporating fishing effort strategies consistent with MSY targets. Correlations between modelled indicators and survey data were significant ($p \leq 0.02$). Reduced fishing effort led to increases in size-based indicators and biomasses of benthivores, planktivores and piscivores. However, predation by piscivores depressed bentho-piscivore biomass. Climate warming may also decrease biomasses of bentho-piscivores and piscivores, while planktivores, benthivores and state indicators of size and trophic level may increase. Fisheries management measures will benefit the biodiversity of the fish community in terms of size structure, but not necessarily the food web since decreases in relative biomass of some trophic guilds are expected.

1. Introduction

Principal objectives for the management of European seas arising from the Common Fisheries Policy (CFP; EC, 2013) and Marine Strategy Framework Directive (MSFD; EC, 2010) are (1) to achieve maximum sustainable yield for all commercial species by 2020 at the latest, while simultaneously eliminating discards, and (2) to achieve Good Environmental Status (GES) of marine waters by 2020 and to protect the resource base upon which marine-related economic and social activities depend. The CFP aims to fulfil its objectives by defining regional fisheries multi-annual management plans that take account of species and fishery interactions in establishing conservation and technical measures to achieve the targets (Articles 9–10).

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http://dx.doi.org/10.1016/j.gecco.2015.06.005

The MSFD reflects a broader governance perspective to support all ecosystem services and considers the management of all human activities that have an impact on the marine environment. In doing so, it captures the objectives of the CFP, which must be achieved within the bounds imposed by the implementation of the MSFD. In the words of the European Parliament and the Council of the European Union, the CFP must "*be coherent with the Union environmental legislation, in particular with the objective of achieving a Good Environmental Status by* 2020" (EC, 2013).

Enshrined in the principle of ecosystem based approaches to management, the assessment of GES is based on evidence from specific indicators that measure attributes of complex ecosystems while accounting for the "prevailing climatic conditions" (EC, 2010). Two examples that are particularly relevant in the context of multi-annual plans in the CFP are food webs and biodiversity (EC, 2010). Although much work is in progress by Regional Seas Conventions (i.e. OSPAR and HELCOM, ICES, 2013a), specific measures and targets for these indicators are not yet internationally agreed. In contrast, fishing mortality targets to achieve MSY for key commercial species (single species F_{msy}) have been identified by the International Council for the Exploration of the Sea (ICES, 2012a) following proposals from the EU Commission (EC, 2006) to implement plans adopted at the 2002 World Summit on Sustainable Development. *In lieu* of certainty or confidence in understanding the dynamics of interactions among species in an ecosystem, an MSY framework is seen as consistent with taking an ecosystem approach to fisheries (Penas, 2007). This is despite the knowledge that efforts to maximise yield of one stock will have an impact on other species and fisheries, because of the interactions among the species and the fisheries that target them (Gislason, 1999; Collie et al., 2003).

These interactions present a major challenge to agreeing management objectives and measures for effective implementation of MSY. They are particularly pertinent in mixed fisheries, where fishing mortality on non-target bycatch species is a consequence of the activity regulated to apply a fishing mortality on a target commercial species (ICES, 2013b, 2015; Mackinson et al., 2009; Ulrich et al., 2012). In addition to direct fishing impacts, both target and non-target species are impacted by changes in abundance of their predators and prey. So, understanding and predicting changes in food webs arising from fishing and climate change is important to the assessment of GES.

Ecopath with Ecosim (EwE) food web models are one of the tools proving useful to evaluate ecosystem impacts of both climate change and fisheries management measures (ICES, 2011, 2015; Lassen et al., 2013). Here, we use a model of the North Sea (Mackinson, 2014) to simulate how biodiversity and food web indicators respond to changes in fishing effort by fleets to achieve MSY fishing. Four model projections are used to compare the strength of fishing and climate influences on the modelled system (Box I), and to reveal trade-offs between ecological components and fisheries that are relevant to making management decisions based on an ecosystem approach.

2. Materials and methods

To model ecosystem effects of fishing at MSY on all fished species groups and their predators and prey, we require a scenario that includes a single fishing strategy that is consistent with the current fishing mortality (F) targets for assessed species. We call this the "fisheries management scenario" and the approach taken to specify this scenario and its associated simulations is summarised in Box I. With this single fishing strategy, in terms of fishing effort, the food web model simulates fishing mortality on all species groups in a consistent way (i.e. not only on species with targets but also on those species without current F targets or unassessed species). The output of the modelled scenario is contrasted to those from a baseline scenario with constant fishing effort and mortality. Forecasts of change in the ecosystem must also account for the potential influence of climate change. So we also generate a climate scenario that continues the trends observed in the data and includes a representation of inter-annual variability based on that previously observed. The output from simulations is used to derive indicators of biodiversity and food webs and thus the response of these indicators is investigated in response to both the fishing impact and the climatic influence. The model, data and methods to determine scenarios to test are described in detail below followed by a description of the indicators that we investigate.

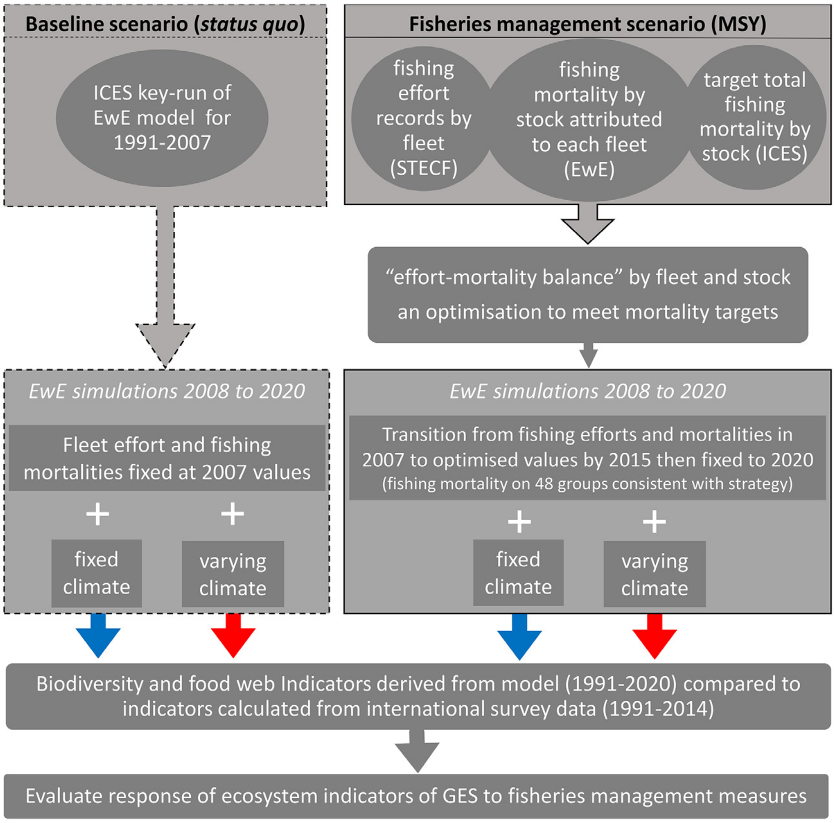
2.1. The multi-fleet food web model

The version of the North Sea Ecopath with Ecosim (EwE) model used here is based on the key-run model reported by ICES (2011), which has been calibrated by fitting to time series data for the period 1991–2007 and includes both fishing and environmental drivers (Mackinson, 2014). To accurately model fishing impacts, the model calibration used instantaneous rates of F, as reported by stock assessments, to drive the model (ICES, 2011). For those species without analytical stock assessments, relative fishing effort data along with estimated catchability values by species (in the Ecopath parameterisation) drive the catch and fishing mortality.

Time series of fishing effort data, arising from the European Commission's Scientific, Technical and Economic Committee for Fisheries (STECF, 2009), are included in the model database for the following aggregated fleets: demersal trawlers and seiners, beam trawlers, pelagic trawlers and seiners, sandeel trawlers, *Nephrops* trawlers, shrimp trawlers. Additional fleets (drift and fixed netters, fleets using hooks, dredges, pots, and 'other') have a fishing impact on some species, but their effort is time-invariant in the EwE key run due to lack of data. The catch compositions of the fleets in the Ecopath key-run parameterisation (based on 1991 catch data) were modified for this study (Appendix A) such that the partial Fs of each fleet at the beginning of the projection year (2008) are a true representation of the data available (STECF 2007 catch data). This ensures that the modelled behaviour of the fleets in future years closely reflects the present situation.

Box I.
Specifying strategic simulations of the multi-fleet food web model of the North Sea to address the question: how will fisheries management measures contribute towards the attainment of Good Environmental Status for biodiversity and food webs?

To model the ecosystem impacts of fishing on 48 species groups in the calibrated North Sea Ecopath with Ecosim model (EwE; ICES, 2011; Mackinson, 2014) we establish two scenarios: a baseline and an alternative given current fisheries management aims. Scenarios are specified in terms of fishing effort values by fleets, which are used to calculate fishing mortalities, F, by species. The "effort–mortality balance" algorithm is used to determine what relative fishing effort values are consistent with fishing mortality targets adopted by fisheries management (ICES, 2012a). The algorithm takes as a starting point the estimated partial F by fleet for each species group in the final year of the ICES key-run of the EwE model along with the reported effort by fleet in that year. Effort levels are then altered by fleet in order to minimise the difference between target Fs and expected total F levels by species. Target values for F are taken from ICES single species assessments and advice and are a combination of MSY (F_{msy}) and long term management plan values (F_{mp}) for 8 assessed species. The model simulations are strategic investigations (*sensu* Plagányi et al., 2014) of the long term impact of fishing strategies on the ecosystem as viewed through the lens of ecosystem indicators.



Note: Fishing scenarios are boxed with dashed (baseline) or solid (fisheries management scenario) lines and this corresponds with the dashed and solid lines in Figs. 2 and 3. Similarly, the red and blue arrows correspond with the colour of the lines for the scenarios forecast in Figs. 2 and 3.

2.2. Target fishing mortality data

The instantaneous target F^s_{target} values by stock, for the optimisation analysis of fishing effort by fleet conducted here (below), were taken from ICES advice for 2012 where possible (Table 1, ICES, 2012a). No suitable F_{msy} values were found for sandeel and sprat: those available in the literature do not reflect the stock and fishery structure in the North Sea, and were not taken further in this analysis. For whiting and cod the long term management plan target for fishing mortality, F^s_{mp}, was considered more suitable than F^s_{msy} since each of these species is being managed towards F^s_{mp}.

2.3. Determining the fisheries management scenario

We determine the fishing effort by fleet required to approach the MSY targets through an optimisation approach (the effort–mortality balance", Box I). The partial[1] fishing mortality F^s_g on species s by fleet with gear g was considered to vary

[1] The percentage of total mortality for each species group split by fishing fleet is given in Appendix A Table S2.

Table 1
Fishing mortalities on EwE species groups, showing final model year value (F 2007), targets for optimisation and optimised values following effort reduction. Also shown are the mean biomass estimates for the modelled species groups and the median catchrate for the surveyed species list (1991–2014). The subset of modelled species groups considered as surveyed species are emboldened. The mean modelled trophic level is the average annual model estimate for the period 1991–2007. Maximum length and guild (Pk = Planktivorous, B = Benthivore, Pi = Piscivore and BP = Bentho-piscivore) are those reported by Engelhard et al. (2011).

Species group	F (2007)	F_{opt} (Target F)	Modelled species-mean biomass (t per km^2)	Surveyed species-median catchrate (t per km^2)	Median annual ratio (biomass/ catchrate)	Mean modelled trophic level	Max length (cm)	Guild
Sprat	0.85	0.70	0.461	0.104	4.771	2.97	16	Pk
Cod (adult)	0.64	0.48 (*0.40*)	0.206	0.139	2.607	4.83	190	Ps
Whiting (adult)	0.44	0.34 (*0.30*)	0.182	0.746	0.281	4.38	70	Ps
Sole	0.43	0.23 (*0.22*)	0.115	–	–	4.00	70	B
Haddock (adult)	0.42	0.31 (*0.30*)	0.167	0.552	0.267	4.26	112	B
Plaice	0.39	0.22 (*0.25*)	0.649	0.041	12.854	3.99	100	B
Herring (adult)	0.33	0.26 (*0.25*)	2.083	0.859	2.552	3.44	40	Pk
Sandeels	0.32	0.32	2.752	–	–	3.35	22.5	Pk
Small demersal fish	0.31	0.30	0.384	0.010	38.552	4.20	–	–
Saithe (adult)	0.25	0.18 (*0.30*)	0.333	0.090	5.456	4.31	120	Ps
Juvenile haddock	0.21	0.16	0.007	0.039	0.267	4.05	112	B
Shrimp	0.14	0.14	0.530	–	–	3.08	–	–
Monkfish	0.12	0.09	0.091	0.007	10.720	4.89	200	Ps
Flounder	0.11	0.05	0.279	0.022	12.220	4.39	50	B
Blue whiting	0.11	0.09	0.093	–	–	4.09	50	Pk
Other gadoids (large)	0.11	0.08	0.072	0.008	7.629	4.59	–	–
Spurdog	0.09	0.07	0.041	0.005	7.807	4.86	105	Ps
Hake	0.08	0.06	0.026	0.002	10.770	4.96	110	Ps
Juvenile cod	0.07	0.07	0.142	0.019	2.607	4.30	190	Ps
Herring juvenile	0.06	0.06	0.094	0.448	0.174	3.43	40	Pk
Large demersal fish	0.06	0.04	0.038	0.001	47.991	4.22	–	–
Catfish (wolf-fish)	0.06	0.04	0.029	0.001	40.716	4.29	125	B
Large piscivorous sharks	0.05	0.05	0.003	–	–	4.95	200	Ps
Miscellaneous filter feeding fish	0.05	0.04	0.036	–	–	3.44	–	–
Mackerel	0.05	0.04	2.910	–	–	3.86	66	Ps
Gurnards	0.04	0.02	0.061	0.153	0.415	4.46	50	B
Horse mackerel	0.04	0.03	0.659	–	–	4.26	60	Ps
Witch	0.03	0.02	0.099	0.002	54.375	4.05	60	B
Turbot and brill	0.03	0.02	0.047	0.003	16.662	4.62	100	Ps
Juvenile whiting	0.03	0.02	0.011	0.133	0.281	4.24	70	Ps
Small sharks	0.02	0.02	0.002	0.001	1.745	4.35	80	BP
Other gadoids (small)	0.02	0.02	0.207	0.009	21.905	3.82	–	–
Norway pout	0.02	0.02 (*0.35*)	0.883	0.258	3.405	3.59	35	BP
Thornback and spotted ray	0.02	0.01	0.070	0.005	15.583	4.53	100	BP
Megrim	0.02	0.01	0.025	0.001	21.848	4.55	50	Ps
Starry ray + others	0.01	0.01	0.088	0.010	7.096	4.47	66	Ps
Dab	0.01	0.01	3.200	0.243	13.078	4.02	40	B
Lemon sole	0.01	0.01	0.330	0.011	31.093	3.94	45	B
Halibut	0.01	0.01	0.026	–	–	4.55	200	Ps
Nephrops	0.01	0.01	1.048	–	–	3.52	–	–
Squid and cuttlefish	0.00	0.00	0.081	–	–	3.84	–	–
Large crabs	0.00	0.00	1.311	–	–	3.85	–	–
Juvenile Saithe	0.00	0.00	0.145	0.003	5.456	3.97	120	Ps
Infaunal macrobenthos	0.00	0.00	141.969	–	–	2.88	–	–

(continued on next page)

Table 1 (*continued*)

Species group	F (2007)	F_{opt} (Target F)	Modelled species-mean biomass (t per km^2)	Surveyed species-median catchrate (t per km^2)	Median annual ratio (biomass/ catchrate)	Mean modelled trophic level	Max length (cm)	Guild
Skate + cuckoo ray	0.00	0.00	0.042	–	–	4.44	92	Ps
Epifaunal macrobenthos	0.00	0.00	80.937	–	–	3.32	–	–
Long-rough dab	0.00	0.00	0.338	0.028	12.933	4.19	50	BP
Dragonets	0.00	0.00	0.045	0.001	40.418	3.99	30	B

linearly in direct proportion to fishing effort by that fleet E_g (Murawski and Finn, 1986). In a simple situation where a single fleet is responsible for the entire fishing mortality on a species, then the scaling factor ε_g of relative fishing effort required to meet the target fishing mortality is equal to the ratio of the target to the observed fishing mortality. However, since fleets catch multiple species and we have multiple targets, the value of ε_g will differ by species. Furthermore, an individual species is caught by more than one fleet such that the scaling factors by species are not independent. Therefore we determine an optimal vector of effort values by fleet with gear g, $E_{g,opt} = \varepsilon_g E_g$, such that the resultant fishing mortality values expected for each species s are as close as possible to the target values. To do this an algorithm was developed to minimise the following objective function (Appendix A):

$$f(\varepsilon) = \sum_{s=1}^{N} \left(1 - \frac{\sum_{g=1}^{G} \varepsilon_g F^s_{g,2007}}{F^s_{\text{target}}} \right)^2 \tag{1}$$

where N is the number of species with target fishing mortalities, F^s_{target}, G is the number of fleets, ε_g is the optimal effort multiplier by fleet and $F^s_{g,2007}$ is the partial fishing mortality on species s by fleet in the final year of the model (Table 1). Once the objective function is minimised (Appendix A) the vector of ε_g can be used to compute the optimal partial and total fishing mortalities from the $F^s_{g,2007}$ values for every fished species in the model. With the targets identified for eight species (cod, whiting, haddock, saithe, Norway pout, herring, plaice, and sole, Table 1), we optimise the fishing strategy using Eq. (1) for the following fleets[2]: Demersal trawler and seiners; Beam trawlers; Pelagic trawlers and seiners.

2.4. Model simulations including fishing and climate scenarios

To run the model forward, two fishing strategies (the baseline "status quo" scenario and the fisheries management scenario) were considered with and without climate variability resulting in four contrasting scenarios (Box I). A gradual transition to fishing at MSY by 2015 was modelled by imposing a scheme where changes in effort, towards the optimised values, occur in steps of not more than a 15% decrease or increase in effort values per year. Climate scenarios are important to consider because primary production and parameters determining the interaction strength among predators and prey in the Ecosim model are conditioned upon the environmental drivers applied in the calibration (ICES, 2011; Mackinson, 2014).

The main driver in the model is sea surface temperature (SST) as measured by the Atlantic Multi-decadal Oscillation (AMO) and the local SST for the North Sea. Additional environmental drivers used are sound speed as a proxy for salinity, dissolved oxygen, and a combined index of the AMO, Hadley SST and total nutrients. To create a climate scenario, a period in the past state of the AMO was defined as a reference period. First, we identified the year in the smoothed AMO (Enfield et al., 2001) in which the most recent minimum occurred (i.e. 1978). We then matched the 'recent segment' of the AMO (from the 1978 minimum to the end of the model period, 2007), to a 'historical segment' of the AMO (i.e. a period of similar length following the preceding minimum), which corresponded to the period 1914–1943. Given the sine like pattern in the smoothed AMO (Cannaby and Hüsrevoğlu, 2009), we assume that future years will begin to repeat this general pattern such that the historical data (1943–1990), once scaled to incorporate the long term trend in the data, can be used to define a future scenario to test the fishing strategies in relation to observed climate variability. In order to retain the structure in the covariance between the environmental parameters used by the model, we assume that values for these other variables are projected similarly. Thus to project the AMO and other variables, the data from the end of the historical segment onward was scaled (such that 1943 values match the 2007 values) and the data from 1944 onward was appended to the end of the recent segment such that a continuous series was created (Appendix A, Fig. S1).

2.5. Indicators of ecosystem effects of fishing

Many indicators have been proposed as candidates for assessments of Good Environmental Status and we can derive a number of biodiversity (BD) and food web (FW) indicators directly from the model simulations. Indicators can be categorised

[2] Exploratory analyses were also made by allowing *Nephrops* trawlers and drift and fixed netters to vary in a restricted manner (limited maximum decrease in effort: $\varepsilon_g \geq 0.75$ respectively (see Appendix A)).

into 'pressure' and 'state' indicators dependent on whether they are derived from human activities on the system or responses of the system, respectively:

(1) pressure indicators: total catch (FW), proportion of primary production required to support fisheries (FW) (Watson et al., 2014), trophic level of the catch (FW) (Shannon et al., 2014) and the associated Fishing in Balance metric (FW) (Pauly and Watson, 2005; Kleisner and Pauly, 2011). Trophic level estimates used here are determined annually from the modelled consumption rates.
(2) state indicators: total biomass (FW), trophic level of surveyed species (FW) (Shannon et al., 2014), the biomass trends of key trophic guilds (FW) (ICES, 2014), Kempton's diversity metric Q modified for use with ecosystem models (BD) (ICES, 2013a), mean maximum length (MML) of demersal fish and elasmobranchs (BD) (ICES, 2012b,c, 2014) and the Large Species Indicator (BD and FW). The Large Species Indicator (LSI) is analogous to the Large Fish Indicator (LFI), Shephard et al. (2012), but based on species biomass data only: the LSI is also known as the 'proportion of large species' (Shin et al., 2010). The LSI and the MML can be considered to represent BD when applied to a single guild (demersal fish and elasmobranchs) and FW when all species are included.

The mean maximum length of fish and elasmobranch species and the trophic guilds considered here (Table 1) are based on previously reported values and classifications (Engelhard et al., 2011). The mean maximum length of fish was also used to separate species into those of large vs small body size for the Large Species Indicator. Following an inspection of the histogram of maximum length (Table 1) two groups were evident: 59% of groups (excluding juvenile groups) were limited by a maximum length of 80 cm and were categorised small, while 41% of groups could reach a maximum length between 92 and 200 cm and were categorised large. Notably, for the Celtic Sea, Shephard et al. (2012) found that the application of a threshold of 85 cm for the Large Species Indicator produced a time-series with a high correlation with the more traditional Large Fish Indicator (Greenstreet et al., 2011).

The diversity index Q (Kempton, 2002) describes the slope of the cumulative species log abundance curve, but has been adapted for use with EwE, whereby the species are replaced by the species groups of the model and the number of individuals by the biomass of the groups (Ainsworth and Pitcher, 2006). The version of the index output from EwE serves largely as an indicator of evenness since it uses the inter-quartile slope of the cumulative abundance curve, does not employ a depletion filter and includes only groups with a trophic level ≥ 3 (Christensen and Walters, 2004). An increase in evenness will manifest itself through a lower slope across the cumulative species log abundance curve and a higher value of the modified diversity index.

2.6. Comparing model derived indicators with survey data

Data from the Quarter 1 International Bottom Trawl Survey (IBTS, (ICES, 2012c)) were used to compare survey based indicators to model output for those indicators suggested by ICES (2013b) as potential indicators for the MSFD (i.e. mean maximum length, trophic level, the biomass trends of key trophic guilds) along with the total surveyed biomass and Large Species Indicator. IBTS data were initially downloaded for 1991–2010 from DATRAS on 15 Jan 2013 and updated for the period 2011–2014 on 29 May 2014 (http://www.ices.dk/marine-data/data-portals/Pages/DATRAS.aspx). Surveys sample species at differing efficiencies, i.e. they are subject to catchability issues, and of course the catch rate units are very different to the biomass estimates from EwE. In order to make the survey data comparable to the model derived indicators we can either 'scale up' the survey data (assume the EwE model biomasses are correct and that the survey captures the whole population of a species such that the scaling factor is the estimated catchability of the survey for that species) or 'scale down' the biomass estimates from EwE (i.e. take a virtual survey from the model). Here, the appropriate scaling factor is computed for each surveyed species (see Table 1 for the species that are surveyed) by estimating the ratio of the average modelled biomass to the average survey catch rate and survey data are 'scaled up'. In order to compare the variable survey data with the model output, LOWESS smooths (2nd degree with 10/length of time series as the span i.e. 0.42) were determined for each survey data time series using R (2013). Finally, smooth indicators were correlated with model derived indicators for identical species groups.

3. Results

3.1. Fishing strategies

The optimised fishing effort multipliers (ε_g), to reach as close as possible to the eight target fishing mortalities simultaneously, were as follows: $\varepsilon_{\text{demersal trawlers/seiners}} = 0.70$, $\varepsilon_{\text{beam trawlers}} = 0.50$ and $\varepsilon_{\text{pelagic trawlers/seiners}} = 0.82$. The optimisation analysis[3] highlighted clear trade-offs between species: while some could be fished close to MSY (haddock and sole) other species were either underfished (saithe, Norway pout and plaice) or overfished (herring, cod and whiting) relative to their target mortality (Table 1 and Fig. 1). No increase in effort for any fleet was suggested so fishing mortality on every group

[3] Simplistic fitting procedures were also implemented to test the algorithm, whereby each fleet's effort was scaled equally (by x such that $\varepsilon_g = x$ for all g) (Appendix A). More complicated exploratory analyses were also made where the algorithm would attempt simultaneously to minimise the overall change in effort and thus a trade-off is incorporated between reaching the mortality targets and affecting the industry. Analyses were also conducted to investigate the benefit in terms of reaching targets of limited change in the effort by *Nephrops* trawlers and Drift and Fixed Netters, but since a large change in effort was required (10% and 9% respectively) with little reduction in fishing mortality (only 0.01 on whiting) this was not considered further.

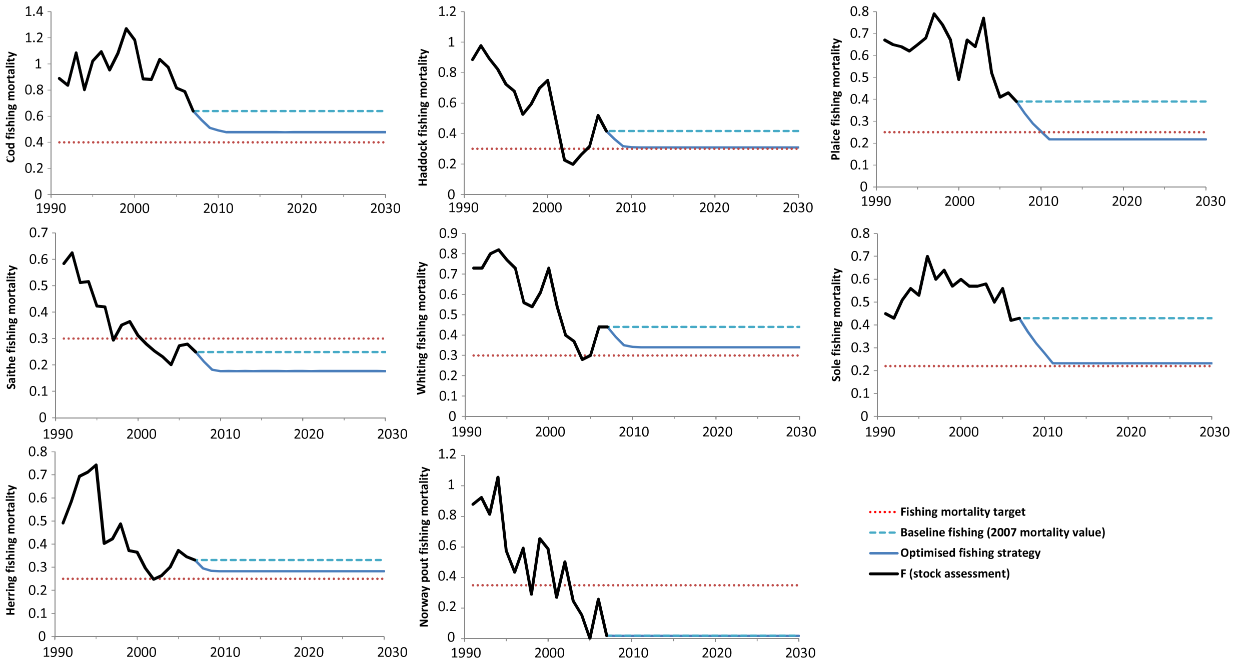


Fig. 1. Fishing mortality model inputs (single species assessment data 1991–2007) with two strategies imposed post 2007. The dashed blue lines (light grey in print) show the baseline status-quo fishing strategy with final year (2007) values of F in the key-run modelled forward and the solid blue lines (dark grey in print) show the optimised Fs (Table 1). The red dotted lines (black in print) show the mortality targets (maximum sustainable or management plan) from single species assessments. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

either decreased or remained unchanged. In the resulting "fisheries management scenario" (Box I), pelagic trawlers and seiners reach optimal effort in 1 year, demersal trawlers and seiners in 3 years and finally the beam trawlers in 4 years. The fishing effects on target species biomasses and yields are shown in Figs. 2 and 3 respectively and the wider effect on all groups within the model is shown in Fig. 4.

Fishing mortality has been reduced greatly since the high levels of the 1980s and 1990s (Fig. 1) and many species (e.g. cod) are already in a recovery phase (Fig. 2). Nevertheless, fishing with the modelled optimal effort by fleet would lead to further decreases in catch of the main species (Fig. 3) and both increases and decreases in the biomass of other species groups (Fig. 2). Plaice and sole benefit greatly from the reduction in beam trawling effort, herring benefit from the decrease in effort by pelagic trawlers and seiners, while cod and to a lesser extent saithe benefit largely from the decrease in fishing effort by demersal trawlers and seiners. No effect of reductions in fishing pressure on the biomass of haddock and whiting was realised and, despite decreases in fishing mortality, the equilibrium biomass of Norway pout was reduced.

A range of winners and losers in the wider fish community was shown due to the effect of reductions in fishing effort (Fig. 4). For example, clear increases in biomass were modelled for sprat and seals. In contrast, decreases were found for megrim, long-rough dab, seabirds, large crabs and dragonets partially through predation and partially due to the reduced levels of discarding expected. Without climate effects, many species require a further 5–7 years to approach their equilibrium values, albeit with some residual fluctuation as in the case of cod (Fig. 2). However, the biomasses of some species take longer to settle down, i.e. haddock (9 years) and Norway pout (12 years) and some species not targeted by the optimised fleets also respond more slowly: in particular long-rough dab, thornback and spotted ray and gurnards also take 11–12 years to approach equilibrium and spurdog slightly longer (14 years). As a result of this optimised fishing strategy (without climate effects), the recovery of cod and stable biomass of saithe drives down haddock biomass and yield through direct predation, despite lower fishing mortality on haddock post 2007 (Figs. 1–3). A similar period of time for the species to reach equilibrium emerges for the baseline case (status-quo fishing) without effort reductions. However, in this baseline case the yields of all species considered for the optimising algorithm are higher (Fig. 3). The depleted species of cod, sole and saithe do not recover greatly in the baseline case and as a result the under-fished stock of Norway pout reaches a higher biomass than in the optimised fishing strategy (without climate effects) (Fig. 2).

3.2. Climate effects

Climate effects in the model are particularly important in terms of both biomass and yields for haddock (positive effects) and whiting, saithe and Norway pout (all negative effects), while herring appear quite robust to both fishing pressure and climate (Fig. 2). Unexpectedly, climate has differing effects on the biomass and yield of cod dependent on the fishing

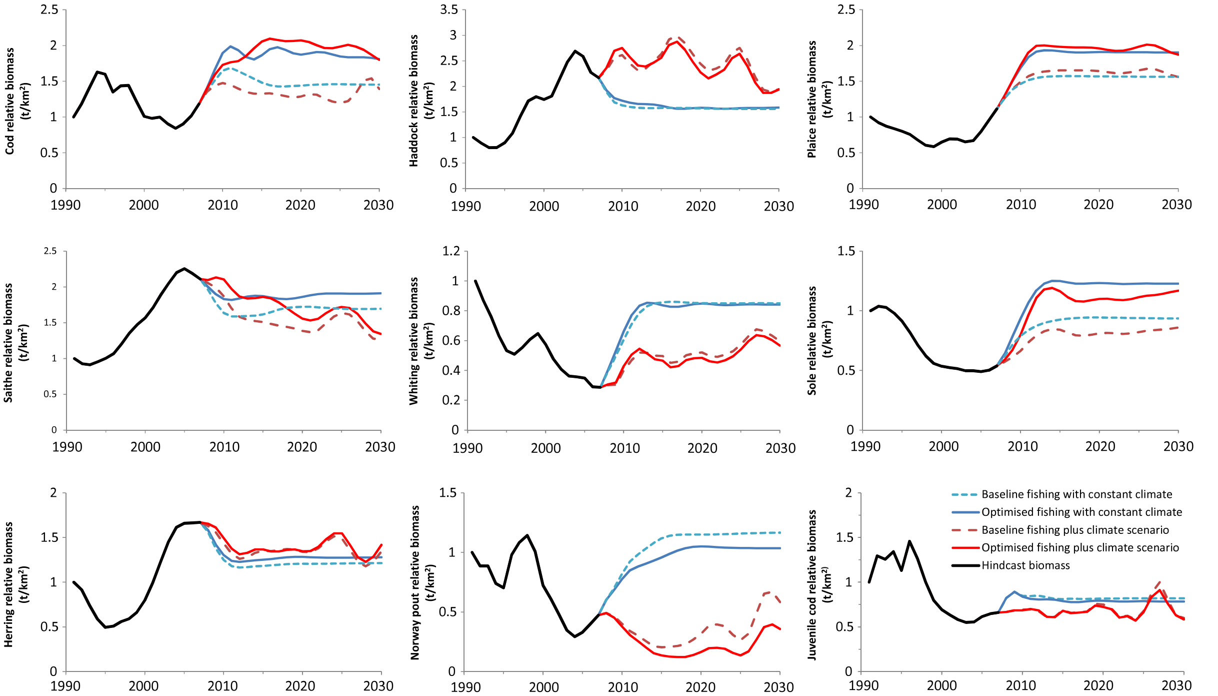


Fig. 2. Modelled time series of biomass, standardised to 1991 values, of the major commercial species. The dashed lines show the relative biomass trajectories following continued fishing under the baseline status-quo scenario (see Fig. 1), and the solid lines show the trajectories with the fisheries management scenario. Blue lines (light grey in print) indicate that the future climate and environmental variables are held constant at 2007 levels, while red lines (dark grey in print) include that the scenario for future climate variability was included. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

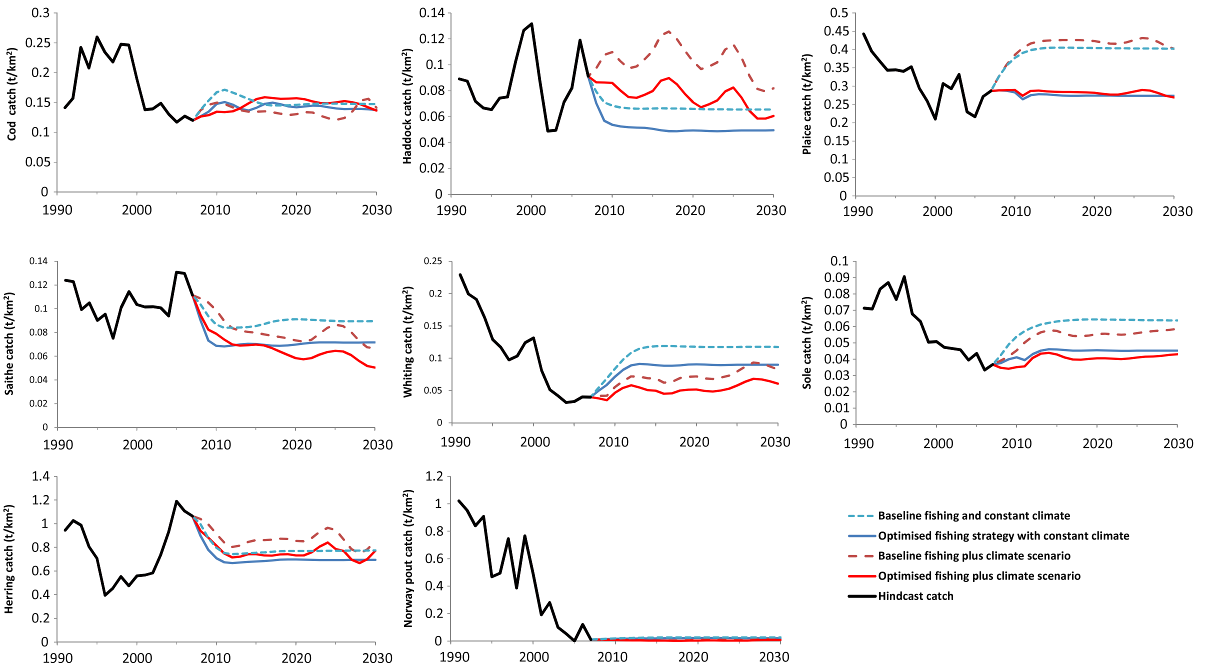


Fig. 3. Modelled yield, standardised to 1991 values, of the major commercial species. The dashed lines show the catch following continued fishing under the baseline status-quo scenario, and the solid lines indicated catch with the fisheries management scenario. Blue lines (light grey in print) indicate future climate and environmental variables are constant at 2007 levels, while red lines (dark grey in print) include a scenario for future climate variability. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

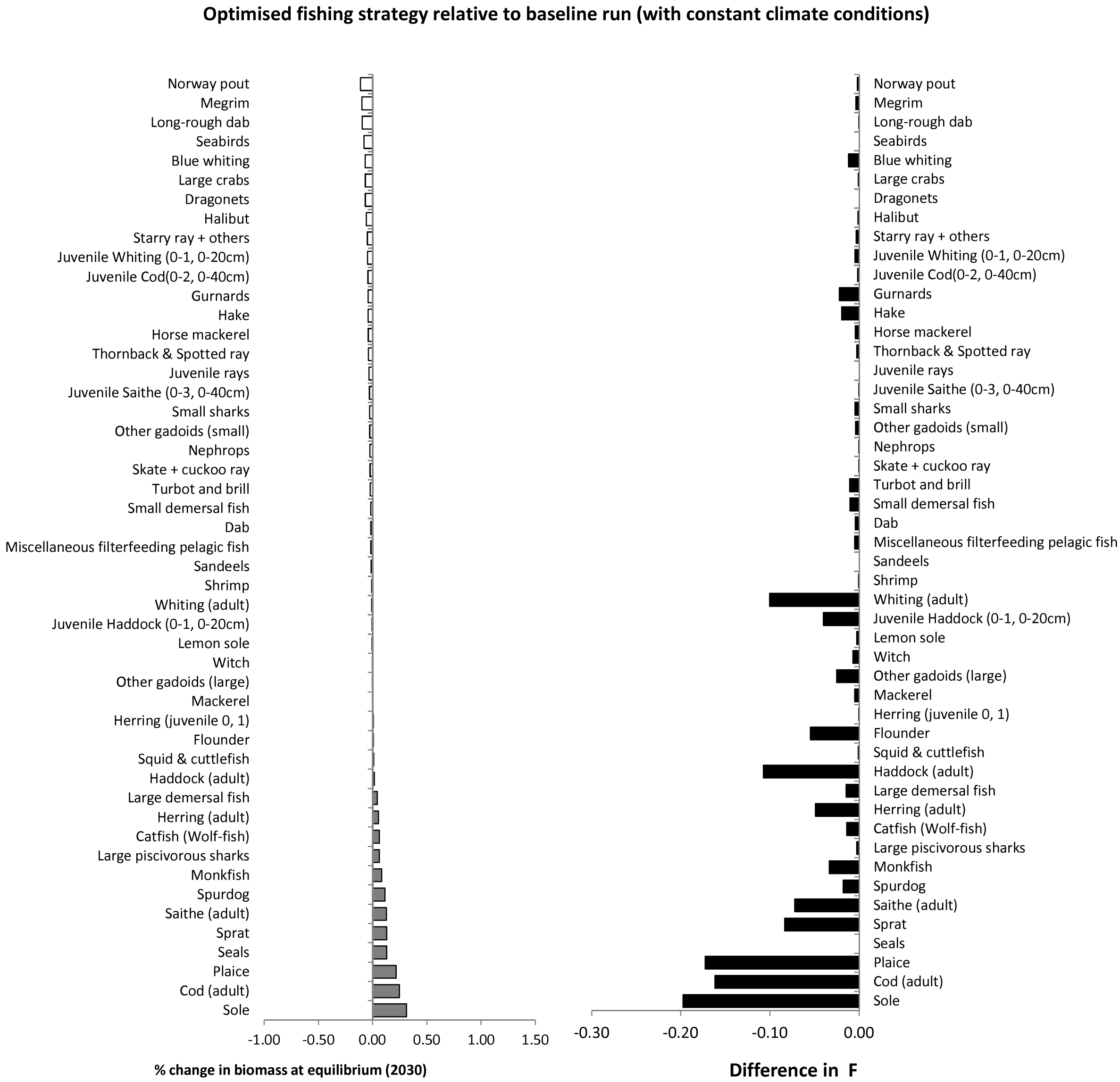


Fig. 4. 'Winners' and 'losers' given the fisheries management scenario under constant climate conditions. Percentage change in biomass at equilibrium i.e. 2030 (left, where increases are shown by grey bars and decreases by white bars) given percentage decreases in fishing mortality (right, black bars) consistent with optimised fishing effort by fleet.

strategy adopted: if the current fishing mortality is sustained the scenario has a negative effect through reductions in the recruitment of young cod, but when fishing effort is reduced the climate scenario results in an increase in feeding success of cod (following rises in prey such as haddock) and a generally positive effect on cod biomass (Fig. 2). For the flatfish, sole and plaice, the climate change scenario had a smaller effect on the change in biomass than the choice of fishing strategy employed. Interestingly though the addition of a non-constant climate scenario had a positive effect on plaice biomass and a negative effect on sole.

Overall the winners/losers in terms of biomass when a non-constant climate scenario was modelled are shown in Fig. 5. Many piscivorous predators such as seals, catfish (wolf-fish), monkfish, spurdog and large piscivorous sharks are predicted to win as a result of effort management and rises in the biomass of their prey and this would appear robust to climate change effects (Figs. 4 and 5). In contrast, among the greatest 'losers', megrim in particular may suffer more greatly given the climate scenario.

3.3. Ecological indicators: fishing and climate effects

As total biomass of the system (including squid, fish, elasmobranchs, seals, whales and seabirds but excluding fish larvae, benthos, epibenthos and plankton) followed the temperature trajectory (Fig. 6(a)), thus the total catch was greater when the

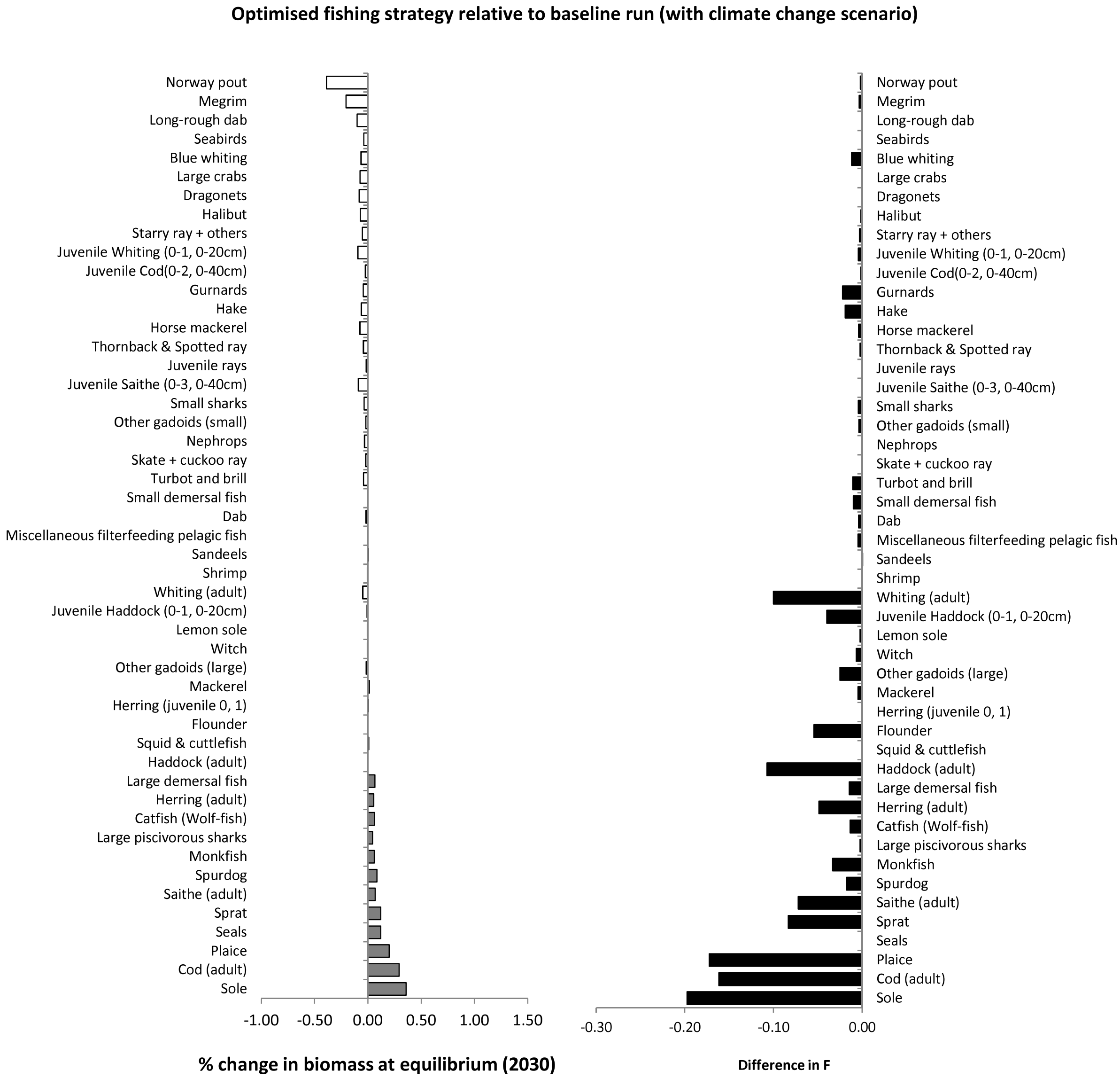


Fig. 5. 'Winners' and 'losers' given the fisheries management scenario and a climate scenario. As Fig. 4 but including the scenario for climate variability. Percentage change in biomass at equilibrium i.e. 2030 (left, where increases are shown by grey bars and decreases by white bars) given percentage decreases in fishing mortality (right, black bars) consistent with optimised fishing effort by fleet.

model was driven by the varying climate scenario than with a constant climate (Fig. 6(e)). The difference in total catch as a result of the reduced fishing effort strategy alone was comparable to the difference in catch between the two climate scenarios: i.e. in fact the difference between the blue lines (fisheries management scenario contrasted with the baseline) in Fig. 6(e) was slightly smaller than the difference between the solid red and blue lines (i.e. the climate effect) with the exception of the final few years of the forecast when temperatures are considered to decrease following the cyclic behaviour of the Atlantic Multidecadal Oscillation. Climate effects on the total biomass and catch are important to incorporate in management scenarios.

The fishing pressure indicators clearly captured the change in the pressure between, i.e. the reductions in fishing effort as a result of the fisheries management scenario relative to the baseline. In the forecast equilibrium state, the trophic level of the catch was reduced (Fig. 6(f)), albeit by a small amount ∼0.05, as was the proportion of primary productivity required to support fishing (Fig. 6(c)). However, the Fishing in Balance index (Fig. 6(d)) also decreased suggesting that the reduced fishing effort scenario was less balanced than the baseline. In the scenarios modelled, the trophic level of the surveyed species rose (Fig. 6(h)) following a decrease in the total catch (Fig. 6(e)): this is largely due to decreases in catch of plaice and sole and subsequent increases in the biomass of each group (Figs. 2–3). Nevertheless, this food web indicator rose by a very small amount since the indicator includes additional species not directly targeted by the fleets optimised here. The trophic level of surveyed species (Fig. 6(h)) appeared more sensitive to the change in the climate scenario than the change in the fishing strategy imposed, due to the great effect of the climate on the total biomass of the system (Fig. 6(a)).

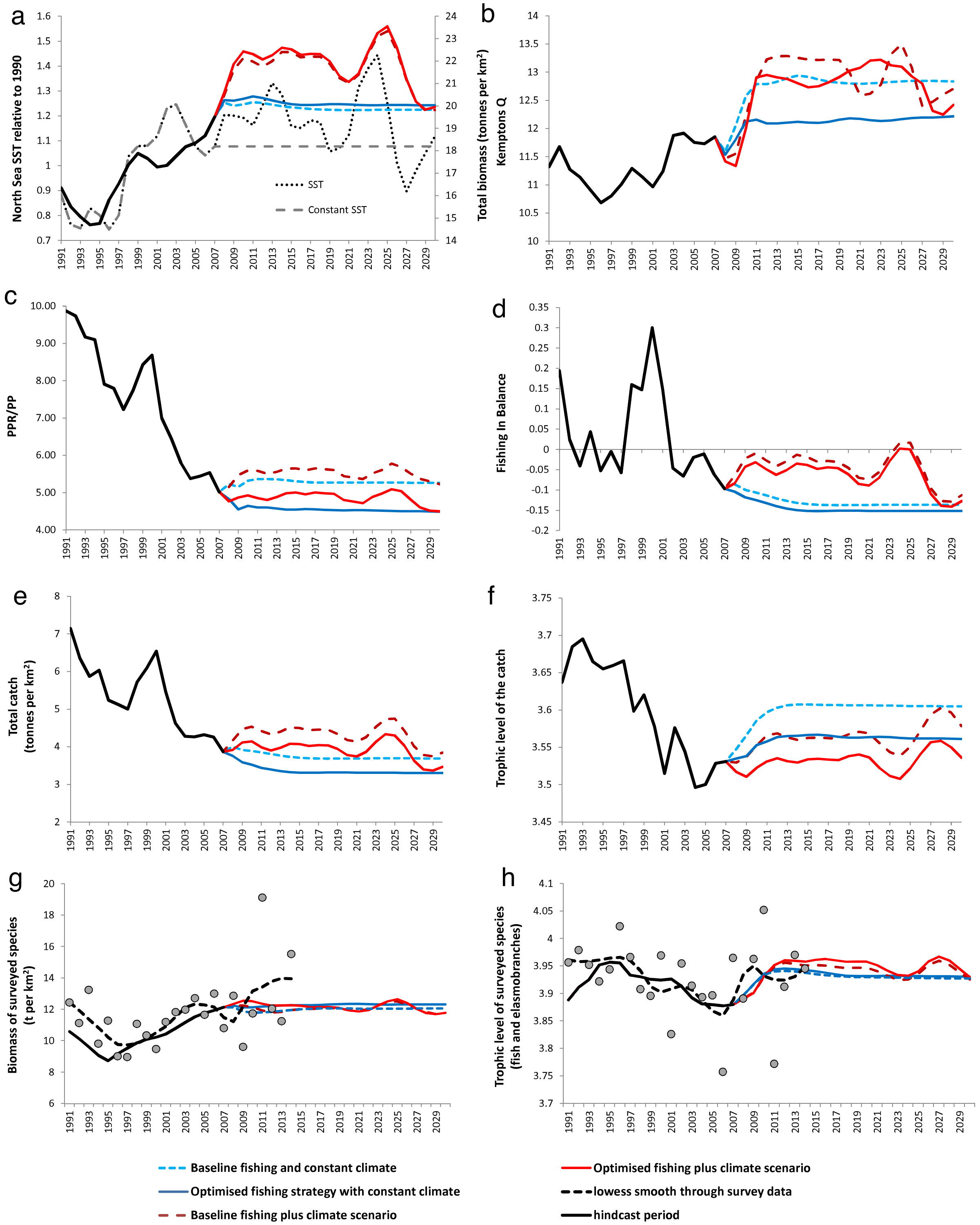


Fig. 6. Modelled derived pressure indicators and ecological indicators with sea surface temperature for each scenario. Pressure indicators shown are: proportion of primary production required to support fisheries (c); the Fishing in Balance metric (d); total catch by all fleets (e) and the trophic level of the catch (f). Ecological indicators shown are: total biomass of the system (a), Kempton's Q (b), biomass of surveyed species (g) and trophic level of surveyed species (h). The sea surface temperature trajectory for each scenario is shown in a, where the dashed grey line shows the constant temperature scenario post 2007 and the black dotted line shows the climate varying scenario. Indicators are coloured by fishing strategy and climate scenario (Box I) as in Figs. 2 and 3. Ecological indicators based on survey data are shown by points with black LOWESS smooth lines. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

The model derived biodiversity indicator, modified Kempton's Q (Fig. 6(b)), showed increases in evenness in all forecasts. However, a comparison of the simulation from the fisheries management scenario with that resulting from the baseline shows a lower level of evenness is attained when fishing effort is reduced, suggesting that species recovering from exploitation attain a more dominant role in the food web. With climate effects included but no change in fishing, Kempton's Q followed the trajectory of the total biomass curve since an increase in sea surface temperatures leads to an increase in lower trophic level groups and their predators (and evenness). When the optimal fishing effort strategy was simulated, the trajectory of Kempton's Q was altered through a restructuring of the food web in terms of relative abundance of trophic groups.

With the varying climate scenario in place, planktivores and benthivores increased in dominance relative to piscivores and bentho-piscivores (Fig. 7(a)–(d)). Interestingly, the temporal change in the planktivore biomass appears closely correlated with change in bentho-piscivore biomass (Fig. 7(b), (d)). However, there is little direct linking between the two functional groups (i.e., a low level of predation mortality by small sharks on planktivores) and the temporal patterns arise from the differing effect of climate on the feeding success of the groups. Reduced fishing pressure alone following the optimal fishing strategy did lead to increases in the biomass of both benthivores and piscivores. However, the increase in piscivores led to a decrease in the biomass of bentho-piscivores through predation.

In the fishing scenario considered, ecosystem effects of fishing were demonstrated clearly by the mean maximum length indicators (Fig. 7(e), (f)) and the Large Species Indicators (Fig. 7(g), (h)) as they both increased as fishing effort was reduced irrespective of climate effects. However, when these two indicators were applied to demersal fish only, as is typically the case, the effect of climate on the indicator was larger. The mean maximum length indicator for demersal fish and elasmobranchs (Fig. 7(e)) increased on average by 5% (2008–2030) due to the inclusion of the varying climate scenario, and this increase was greater than the 1% increase when all species were included (Fig. 7(f)). Similarly, the Large Species Indicator for demersal fish and elasmobranchs (Fig. 7(g)) increased on average by 10% (2008–2030) when a varying climate scenario was included, nearly double the increase relative to that attained when all species were included (6%) (Fig. 7(h)).

3.4. Ecological indicators: model versus survey data

Ten state indicators were evaluated against survey data: i.e. total biomass and trophic level of surveyed species, mean maximum length (two variants), Large Species Indicator (two variants) and the biomass of four trophic groups (Figs. 6 and 7). Each correlation between survey and modelled indicators was significant (all $R \geq 0.47$, $df = 22$, $p \leq 0.02$) for the period 1991–2014 including the forecast 2008–2014 based on the varying climate scenario and constant fishing mortality. The poorest modelled indicators were the biomasses of piscivores and bentho-piscivores, trophic level of surveyed species and both variants of mean maximum length ($0.47 \leq R \leq 0.63$, the remaining five indicators had R in the range $0.75 \leq R \leq 0.87$ with $p < 0.0001$). The two guild indicators (piscivore and bentho-piscivore biomass) showed stronger correlation when the modelled indicators were constructed using the constant climate in the forecast period 2008–2014 (from $R = 0.47$ to 0.62 for piscivore and $R = 0.63$ to 0.88 for bentho-piscivore) as did the trophic level indicator ($R = 0.60$ to $= 0.67$) suggesting that the model has not captured all important climate processes for piscivores and bentho-piscivores (Fig. 7).

4. Discussion

Our results highlight that it is not possible to fish all species at their target mortalities simultaneously since many species are caught by a range of fleets such that management of one fleet has implications for another. For some species to be fished at or near the mortality targets, consistent with maximum sustainable yield (MSY), other species will be either overfished or underfished in terms of fishing mortality. Given also that fish consume one another, the total catch of one species will affect that of others through knock-on effects in the food web. These simple facts mean that predicting how changes in fishing effort influence long-term yields in mixed fisheries requires explicit consideration of the interactions among species and the fleets that target them (Dickey-Collas et al., 2014). Likewise, evaluating the ecosystem effects of fisheries can only be achieved by understanding how the combined effort of fishing fleets impact upon the variety of target and non-target species that are caught or affected indirectly through the food web or impacts on their habitat (Engelhard et al., 2014).

4.1. Trade-offs in mixed fisheries and multi-species effects

In the mixed-fisheries context, fishing for saithe at the single-species target rate is not attainable without over-fishing cod, haddock and whiting (Table 1). Similarly, fishing for Norway pout and herring at their target levels simultaneously is not possible and a compromise emerged here between overfishing herring and under-fishing Norway pout (Table 1). However, these trade-offs are only one part of the puzzle since the species also interact with each other and their biomasses are influenced by climate processes. In the scenarios modelled here, the biomass of Norway pout falls in response to predation by saithe and also due to climate effects, despite fishing rates on this species being much below the MSY target. As a result, predators such as megrim with a large proportion of its diet composed of Norway pout (Mackinson and Daskalov, 2007), are also predicted to decline. Similarly, climate effects are particularly important for haddock and the modelled scenarios indicate that bottom-up forcing of this stock can be more important than the fishing pressure exerted by the fleets.

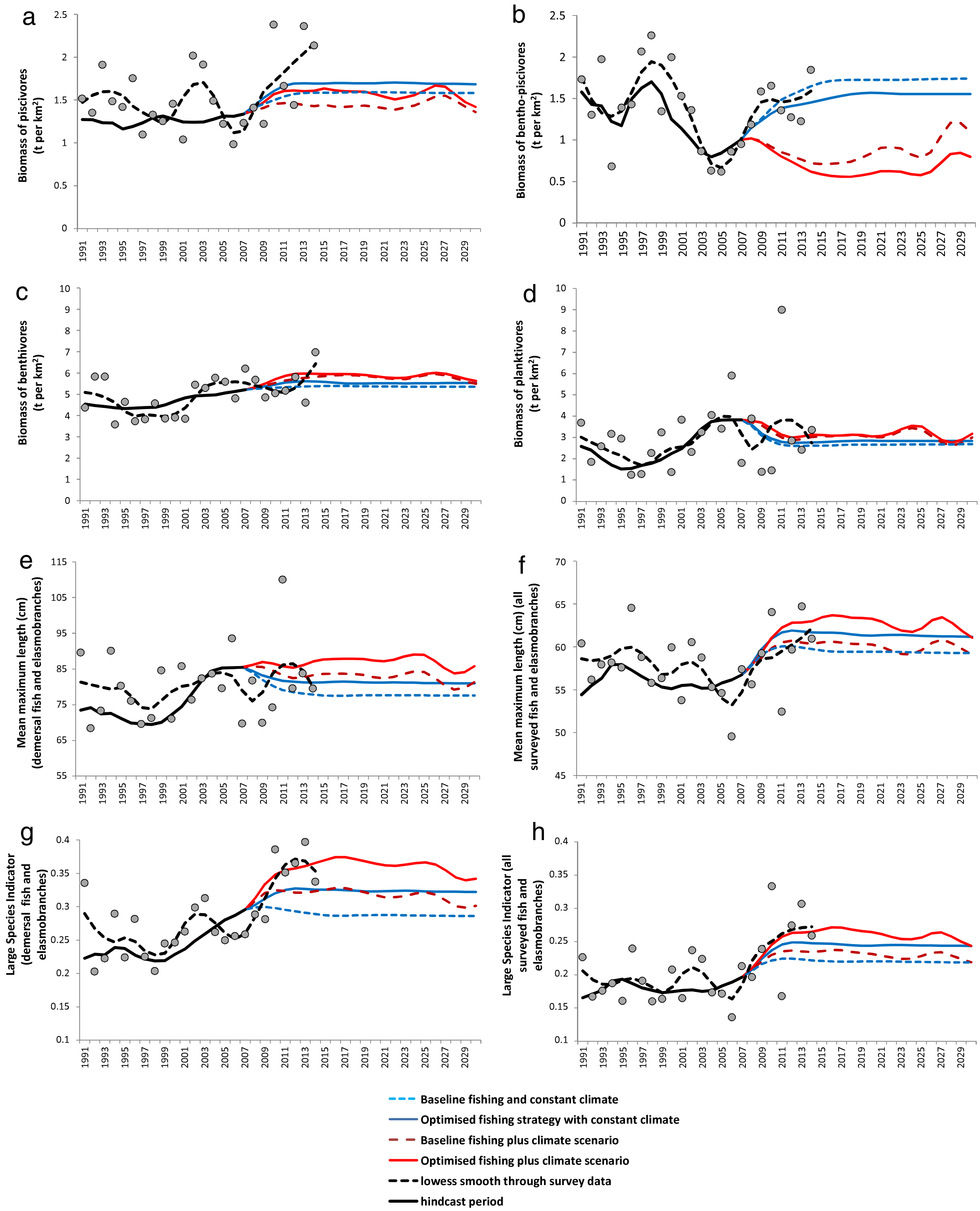


Fig. 7. Ecological indicators for each scenario: biomasses of key trophic groups of fish: piscivores (a), bentho-piscivores (b), benthivores (c) and planktivores (d); mean maximum length of fish (e) and elasmobranchs (g) for demersal species (left) and all surveyed species (f, h, right). Lines are coloured by fishing strategy and climate scenario (Box I) as in Figs. 2 and 3. Ecological indicators based on survey data are shown by points with black LOWESS smooth lines. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

What this research points to is the strategic value of multi-fleet food web models in evaluating possible risk of over/underfishing relative to targets, rather than implying confidence in the precision of predictions that may be required for setting tactical management rules (Plagányi et al., 2014). Here, we acknowledge the large uncertainties that arise from uncertainty in model parameters. Current work is underway to take this into account by providing outputs derived from simulations where parameters' uncertainties are explicitly considered. Additionally, work in progress by the IndiSeas working group may serve to generalise the findings made here for the North Sea ecosystem (Shin et al., 2012).

We recognise that it is not wholly adequate to assume that the targeting of fleets remains the same from year to year, such that differences in their impact are determined solely by changes in effort. Ideally we would have data on changes in effort of all the fleets in the model and, ideally, the catch compositions of modelled fleets should depend upon harvest control rules and a behaviour model where fishing choices depend on the balance of costs and gains. Such dynamic fleet behaviour models are necessary to more accurately forecast changes in catch and fish mortality. However, the complexity of fleet behaviour (Tidd, 2013) has thwarted efforts to achieve this. Other models that attempt to partition mortality among fleets (e.g. ''*F*-cubed'', used in ICES mixed fisheries advice) suffer from the same problem: the need to assume that fleets behave the same way in the future as in the past. Our goal here is to expose the ecological and fishery trade-offs arising from alternative policy choices, so the model needs to reflect the relative differences in mortality among fleets at the 'time' the policy is introduced. Here we show that even with effort reductions aimed at fishing as close as possible to F targets some species (e.g. haddock) will decline in biomass due to predator–prey interactions.

Our pragmatic approach to determining a future climate scenario allows us to represent in a realistic way the variability in environmental drivers that has been observed in the past. Some caveats are required, for instance some historical changes in nutrient levels due to changes in human activities will also be replicated in the future. Nevertheless, the main environmental driver of change in the model is temperature and this approach provides a simple scenario that captures the main warming signal for the period to 2030 with a realistic level of variability included. However, this climate scenario is not meant to be taken as a prediction of future climate.

4.2. Quantifying wider ecosystem effects

A range of indicators have been developed by the scientific community (Kleisner et al., 2015; Shannon et al., 2014; Shin et al., 2010, 2012) and novel modelling techniques have been proposed to address relationships between pressure and state indicators (e.g. Fu et al., 2015). Currently, the OSPAR Convention is considering the mean maximum length indicator as an indicator of biodiversity for demersal fish, while the trophic level of surveyed species and the abundance of key trophic groups (e.g. planktivores, piscivores, benthivores, bentho-piscivores) are considered in terms of food web indicators (ICES, 2013b). Additionally, the Large Fish Indicator (approximated here by the Large Species Indicator) is considered as both an indicator of biodiversity and an indicator of food webs, dependent on whether or not the index is computed from demersal species only or both demersal and pelagic species.

In the model forecast period post 2007, the majority of pressure indicators fall and the majority of state indicators rise (Figs. 6 and 7). The notable exception is the state indicator for the biomass of the planktivore guild (Fig. 7(d)), which falls in the period 2007–2011 contributing to the rise in the pressure indicator, TL of the catch (Fig. 6(f)), since the dominance in the catch shifts towards piscivores (Fig. 7(a)). When climate variability is included in the simulations the rise in piscivores and decline in planktivores lessen and the biomass of bentho-piscivores falls, such that the TL of the catch does not increase greatly. After a period of ∼10 years, all indicators reach equilibrium. The model simulations demonstrate that the North Sea food web is recovering following the reduction in fishing pressure on commercial species: total biomass increasing (Fig. 6(a)), catch decreasing (Fig. 6(e)) and proportion of primary production required to support the catch decreasing (Fig. 6(c)). However, this necessarily implies that some species will fare well while others will suffer through increased competition and predation (as can been seen from the species level changes in biomass and the relative biomass of trophic guilds, Fig. 7(a)–(d)). The modified Kempton's Q (Fig. 6(b)) did not respond in a simple way to either fishing or climate, yet the metric did integrate change within the food web in a different way to the other indicators. Further investigation of modified Kempton's Q is required to understand its response to change in food webs. In particular, further testing is required to understand its sensitivity to the trophic level cut off used for inclusion of species groups into the dataset.

In the ecosystem studied, reductions in fishing effort (as implemented in the scenario tested) towards MSY targets for fishing mortality should lead to further increases in the indicators of size structure (mean maximum length and the Large Species Indicator, Fig. 7(e), (g)) and the total biomass of fish and elasmobranchs (Fig. 6(a)), largely due to increases in the biomass of piscivores and benthivores (Fig. 7(a), (c)). These increases in the North Sea may be further fuelled by rises in the primary production of the system driven by increases in sea surface temperature (Fig. 6(a)). As a result of the recoveries in these components, the biomass of bentho-piscivores is expected to decrease (Fig. 7(b), (d)). The trophic level of surveyed species integrates these effects and shows little change on average in the time series (Fig. 6(h)).

Overall, the proposed reductions in fishing effort alone did not alter the total biomass of the modelled system (Fig. 6(a)) despite decreases in total catch (Fig. 6(e)). However, the reductions in effort did lead to a smaller rise in evenness of the higher trophic levels (Fig. 6(b)), due partially to an increase in benthivores (e.g. Catfish/Wolf-fish, Fig. 7(c)) relative to the less abundant bentho-piscivores (e.g. Long-rough dab, Fig. 7(b)) and a clear increase in the mean maximum length of fish (Fig. 7(e), (f)) and Large Species Indicator (Fig. 7(g), (h)). Climate effects had a larger effect on the relative biomass of trophic groups than the reductions in fishing effort (e.g. increase in planktivores, Fig. 7(d), due to increases in primary production, but decreases in bentho-piscivores, Fig. 7(b)), such that evenness of the system was elevated and particularly so in the low fishing effort regime (Fig. 6(b)).

5. Conclusions

Fishing mortality targets based on single species assessments and a combination of MSY considerations and management plans have been set for numerous European fish species. Here, we show how to determine a fishing strategy objectively by optimising the effort of different fishing fleets in order to balance trade-offs between over-fishing and under-fishing species relative to mortality targets. We further show that when these strategies are applied in a food web model, predator–prey interactions and climate change can alter greatly the expected biomass of some species and thus the fishery yields (Mackinson et al., 2009). Research is on-going to identify suitable targets for F in a multi-species context to account for predator–prey interactions (Ulrich et al., 2008, 2011) and further research is also required to demonstrate whether such management aims are appropriate given the requirement to meet GES by 2020 for all descriptors of the MSFD.

In the scenarios investigated here, both the mean maximum length indicator of biodiversity and the Large Species Indicator (as either biodiversity or food web indicators) responded consistently to fishing (i.e. the indicators maintained higher levels under a reduced fishing effort strategy relative to the baseline "status quo" in each climate scenario). Although these indicators were also responsive to climate effects in the scenarios, the influence of climate can be reduced through the broadening of the species list for the indicator to include pelagic species. The trophic level of surveyed species was responsive to climate with a marginal effect of fishing only. Modelled temperature effects suggest that the biomasses of piscivores and bentho-piscivores may be suppressed by warming and, if not taken into account during target setting, targets for these indicators could conceivably be missed due to climate effects.

So, how will fisheries management measures contribute towards the attainment of Good Environmental Status for the North Sea ecosystem? Measures reducing fishing effort that lead to the implementation of fishing at MSY levels would serve to increase the mean maximum length and large species indicators of fish and elasmobranchs and the biomasses of piscivores, planktivores and benthivores. However, there will be associated decreases in bentho-piscivores and although community evenness will rise it will be less even than is possible at higher fishing pressure.

For an indicator based assessment of environmental status, a selection of indicators, including both pressure and state indicators, is required to decipher the underlying patterns in the system. Fisheries management must consider environmental targets for indicators alongside traditional fisheries mortality targets such that they are not in conflict and ecosystem modelling can facilitate this process.

Acknowledgements

The authors would like to thank Carl Walters for informative discussions on the internal calculations made by the Ecopath with Ecosim software and also Georg Engelhard and Axel Rossberg for reviewing a draft manuscript. This work was made possible through funding from UK Defra project Fizzyfish (M1228), DEVOTES EC FP7 Grant Agreement 308392, GAP2 EC FP7 Grant Agreement 266544, and MYFISH EC FP7 Grant Agreement 289257.

Appendix A. Supplementary data

Supplementary material related to this article can be found online at http://dx.doi.org/10.1016/j.gecco.2015.06.005.

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